Factors Affecting Restoration of *Halodule wrightii* to Galveston Bay, Texas

Pete Sheridan[1]
Garry McMahan[1]
Kamille Hammerstrom[1]
Warren Pulich, Jr.[2]

Abstract

Experimental restoration of *Halodule wrightii* (shoalgrass) to its former range on Galveston Island, Texas, began in April 1994. We tested the effects of site, planting density, water depth, and fertilizer addition on survival and growth through June 1996. Temperature, salinity, light transmittance, turbidity, and sediment properties at two restoration sites, Redfish Cove and Snake Island Cove, were similar to those in naturally occurring grassbeds in nearby Christmas Bay. *Halodule* survival, coverage, and new shoot densities were affected by site (significantly higher at Redfish Cove than at Snake Island Cove, which eventually failed), by planting density (significantly higher when planted on 0.25-m or 0.5-m centers rather than on 1.0-m centers), and by water depth (significantly higher when planted in relatively shallow water). Propagation (spreading from transplant units) was significantly greater from 0.25-m or 0.5-m center plantings but was not consistently affected by site or water depth. Fertilizer enhanced propagation but not survival. After two years, Redfish Cove produced belowground biomass similar to that observed in Christmas Bay, but aboveground biomass remained significantly less. Snake Island Cove plant mortality in September 1995 may have been presaged by low root-rhizome carbohydrate lev-

els observed in October 1994, but causes remain unknown. Further restoration of *Halodule* to Galveston Bay is possible at selected sites, but structural equivalency will take longer than two growing seasons to achieve.

Introduction

Seagrasses, primarily *Halodule wrightii* (shoalgrass), disappeared from the margins of West Bay, the western arm of the Galveston Bay estuary of Texas, between 1975 and 1982 (Pulich & White 1991; Hammerstrom et al. 1998). This loss of over 450 ha was attributed to the direct and indirect effects of waterfront dredging and wastewater discharges (Pulich & White 1991) or to unidentified point-source pollution (Adair et al. 1994). Such factors have caused declines in seagrass acreage along the margins of northern Gulf of Mexico bays by 30–80% over the past four decades (Duke & Kruczynski 1992). Seagrasses such as *Halodule*, *Thalassia testudinum* (turtlegrass), *Halophila engelmanni* (clovergrass), and *Ruppia maritima* (wigeongrass; spelling of common name following Kantrud 1991) still grow in the Christmas Bay state coastal preserve, a semi-isolated embayment southwest of West Bay (Pulich & White 1991; Adair et al. 1994).

The Galveston Bay Comprehensive Conservation and Management Plan calls for the restoration of 567 ha (1400 acres) of submerged aquatic vegetation by the year 2004 (Galveston Bay National Estuary Program 1995). This habitat serves to stimulate local productivity, and seagrasses usually support higher densities of fishery and forage organisms than do nonvegetated sands or muds (Fonseca 1992; Fonseca et al. 1996b). Several recent reports indicated that current environmental conditions are suitable to attempt seagrass restoration in West Bay. Water and sediment quality have improved since the 1960s: wastewater discharges and total suspended solids have been reduced, and pollutant inputs such as metals and petroleum byproducts have been controlled (Ward & Armstrong 1992), although certain estuarine areas still have the potential for degradation (Carr 1993). In addition, waterfront development of canal communities on western Galveston Island has abated, maintenance dredge materials must be disposed in uplands, and preliminary experiments indicated that restoration of formerly vegetated shorelines with *Halodule* and *Ruppia* is possible (Hammerstrom et al. 1998).

The goal of this project was to determine whether small-scale restorations in West Bay resulted in viable *Halodule* habitat. The objectives were to evaluate the suitability of the sites in terms of water and sediment quality to determine the survival, growth, and persis-

tence of *Halodule* transplants, and to compare structures of restored and adjacent natural beds. Measurements of water temperature, salinity, light transmission, and sediment concentrations of metabolically required and contaminant trace metals were made in transplanted and natural beds and were compared with published criteria (e.g., minimum light requirements) or, if criteria did not exist, to conditions typical of natural seagrass beds elsewhere in Texas. In addition, several criteria typically used to judge transplant success (survival and propagation of planted material, coverage, shoot density, biomass) or suggested as indicators of seagrass health (chlorophyll and carbohydrate contents, elemental composition) were monitored and compared to those in a natural seagrass bed. These characteristics were used to determine whether the large-scale restoration planned for Galveston Bay could be successful.

Methods

Suitability of Restoration Sites

Restoration was attempted at two sites on western Galveston Island (Fig. 1). One bed was planted at Redfish Cove (29°06′15.9″N, 95°06′30.9″W), and two beds (East and West) were planted at Snake Island Cove (29°09′21.4″N, 95°01′58.5″W). Restored beds were compared with seagrasses along three transects in undisturbed grassbeds of nearby Christmas Bay (East transect, 29°02′48.3″N, 95°10′14.8″W; West 1 and West 2 transects,

Figure 1. Experimental and natural seagrass sites in West Bay, part of the Galveston Bay estuary.

29°02′26.1″N, 95°10′52.6″W). Historical data and preliminary sampling in 1993 indicated that salinity, turbidity, sediment organic content, and sand-silt-clay proportions of sediments were similar at all three sites (K. P. Hammerstrom et al. 1998; G. Conley, Texas Natural Resources Conservation Commission, Beaumont, Texas, personal communication).

Continuously recording temperature, salinity, and light meters were deployed at the midpoint of the eastern edge of the Snake Island East bed and near the middle of the Christmas Bay West 1 transect immediately prior to restoration activities and through the first growing season (20 April–3 November 1994). No meters were placed at Redfish Cove. Submerged temperature and salinity probes (Datasonde 3, Hydrolab Corp., Austin, Texas) recorded data at 30-minute intervals except when they were removed for servicing or following power failures, and means and ranges were compared with previously published tolerance limits for *Halodule* in Texas (Simmons 1957; McMillan & Moseley 1967; McMahan 1968). Point estimates of temperature, salinity, and turbidity were also recorded during site visits after restoration to verify data sonde records and to compare the first and second growing seasons when data sondes were not available. Temperature and salinity were measured by thermometer and temperature-compensated refractometer on arrival and departure during site visits. Water samples were collected at these times to determine turbidity (nephelometric turbidity units, or NTU; HG Scientific Model DRT100B turbidimeter).

Photosynthetically active radiation (PAR, 400–700 nm) was measured with paired quantum sensors attached to data loggers (LI-COR, Lincoln, Nebraska) following the methods of Dunton (1994). A flat quantum sensor (Model LI-192SA) was mounted on top of each data logger housing 2 m above the sediment surface to measure incident light. An owl replica was placed atop a 2-m-tall post adjacent to each housing to prevent interference with the flat sensors by roosting birds. A spherical quantum sensor (Model LI-193SA) was attached to a submerged stake 10 cm above the sediment to measure transmitted and reflected light within each seagrass bed. The submerged sensor was placed 2 m away from the data logger housing to avoid shadows. Light sensors were cleaned every 3–4 days to remove sediments and biofouling that reduced sensor accuracy. Photon flux density (PFD, μmols/m^2/sec) was integrated over a 1-minute period each hour, multiplied by 60 to estimate hourly totals for each hour per day that PAR was detected, then summed to estimate daily total PFD (mols/m^{-2}/day). Light transmittance (%) to the bottom was calculated by [100 × PFD(submerged)]/PFD(incident). Continuous recordings were made except when meter arrays were detached from their

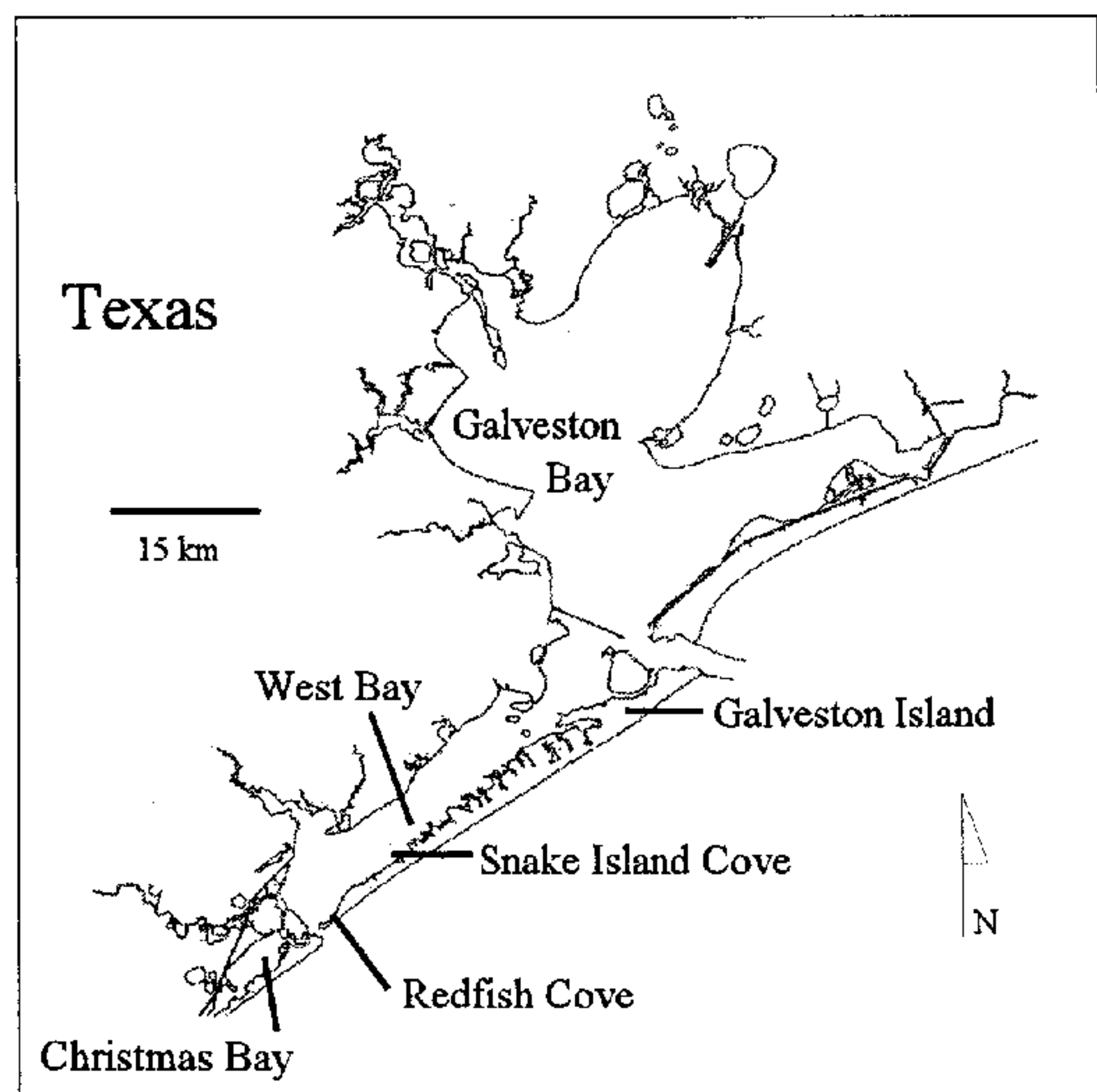mounts for servicing or following power failures. Total underwater PFD and mean transmittance were compared to minimum light requirements for *Halodule* established by Dunton (1994).

Sediment samples for trace metal analysis were taken from the two restoration sites and from Christmas Bay in October 1994. Sediments were analyzed for metals required in seagrass metabolism (Fe, Mn, Zn) and for toxic contaminants (As, Cd, Cu, Ni, Pb; Pulich 1980). Four replicate cores (5 cm diameter × 10 cm deep) were taken at regular intervals diagonally across each site at Redfish Cove and Snake Island East and along the Christmas East transect. Cores were sectioned into depths of 0–5 cm and 6–10 cm. Subsamples of sediment from the interior of each core section were placed in separate acid-washed glass jars and sealed with para-film-lined lids. Sediment analyses followed the methods of Pulich (1980). Wet sediments were thoroughly homogenized, and subsamples of approximately 3.0 g were digested in boiling aqua regia for three hours to extract metals. Digestates were diluted to a final volume of 35 ml and centrifuged. Flameless atomic absorption spectrophotometry was performed on the supernatant by means of a graphite furnace to detect arsenic, cadmium, copper, nickel, and lead, whereas flame atomic absorption was used to detect iron, manganese, and zinc. Sediment water content was calculated by weighing another subsample before and after drying at 100°C for 72 hours. Metal concentrations were compared to values characteristic of apparently healthy *Halodule* beds experiencing a gradient of industrial contamination of sediments (Pulich 1980).

Several quality control tests were run simultaneously with the metal analyses: blank determinations, duplicates of three of the samples, matrix spikes, and blank spikes recovery. Blanks for all metals were much less than 0.1% of sample values, except for arsenic and lead, which showed measurable backgrounds of 3% and 1.25%, respectively. Blank spike recoveries were as follows: 81–117%, arsenic; 73–94%, cadmium; 84–106%, copper; 94–103%, manganese; 72–122%, nickel; 82–94%, lead; and 74–96%, zinc. Only one sample appeared aberrant (36% recovery of lead).

Halodule Restoration

Halodule transplanting units (TPUs) were collected from donor beds in Matagorda Bay near the towns of Palacios and Port O'Connor, Texas, approximately 15 km apart and 150 km southwest of West Bay. A total of 3500 TPUs was removed from the Palacios bed and transported to Redfish Cove during 26–28 April 1994. Another 7100 TPUs were removed from the Port O'Connor bed and transported to Snake Island Cove during 3–5 and 10–12 May 1994. TPUs were prepared by the peat pot method developed by Fonseca (1994). A 7.5-cm-diameter circular sod plugger was used to extract *Halodule* from donor beds. Each plug was then inserted directly into a 7.5-cm-diameter peat pot. Time-release fertilizer (Osmocote: 14% N, 14% P, 14% K; 5.25 g ± 0.14 SE, $n = 10$) was added to all but 1200 TPUs from Port O'Connor prior to insertion of *Halodule* plugs. TPUs were placed in seawater-filled holding tanks and covered with wet burlap to prevent desiccation during transport. Once at the transplant site, the sod plugger was used to create holes into which TPUs were placed, after the sides of the peat pots were ripped to allow rhizome propagation (spread from the original TPU).

Individual transplant beds encompassed 62 m × 42 m and were arrayed perpendicular to the shoreline at the low-water mark observed after passage of a strong cold front in December 1993. The perimeter of each bed was marked by galvanized fence posts every 2 m and was surrounded by 1.2 m-high black plastic screen with 10 mm × 16 mm mesh during the first week of April 1994. This mesh size was expected to exclude all but the smallest of the local fishes and decapods, several species of which could affect transplant success through disturbance or herbivory (Fonseca 1994; Hammerstrom et al. 1998). Access gates were built in two corners. After fencing, each bed was swept from the seaward edge to the landward gate with a 75 m-wide, 10 mm-mesh seine to force large organisms out, then the gate was closed. Gates permitted re-entry of transplant and monitoring personnel but not large aquatic organisms. By September, abrasion from oysters and barnacles on fence posts had created large holes in the screens, so screens were removed in October 1994.

Snake Island East and West were constructed 20 m apart and 9 km northeast of Redfish Cove. Each bed received TPUs arranged in three 8 m × 54 m strips (Fig. 2). Within each strip, we delineated areas representing combinations of three planting densities (high = 0.25 m centers; medium = 0.5 m centers; low = 1.0 m centers) and three relative water depth ranges (shallow, medium, and deep). Walkways were left between sections, between strips, and along the inside edges of the fence. Patterns were repeated in each bed for ease of construction, with two exceptions in the Snake Island West bed: (1) the strip with 0.25 m centers in shallow water was switched with that containing 0.25 m centers at mid-depth to test for possible edge effects on seagrass survival and growth; and (2) all TPUs in the strip with 0.25 m centers in deep water received no fertilizer to allow comparison of seagrass survival and growth with that of other plugs receiving fertilizer. Within each strip, TPUs were planted along marked guidelines as follows: 40 rows at 9 TPUs per row on 1.0 m centers (360 TPUs); 24 rows at 17 TPUs per row on 0.5 m cen-
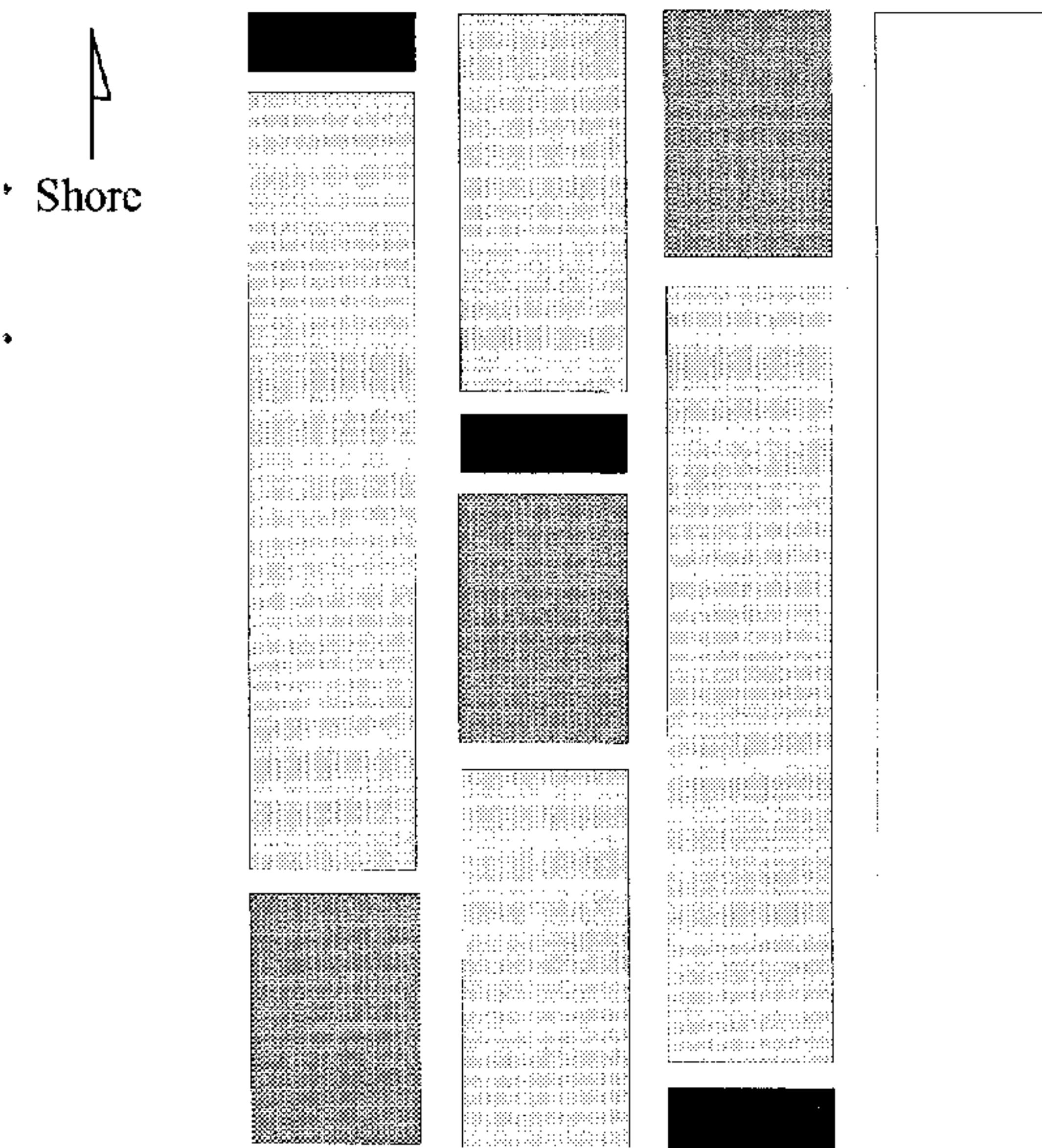
Figure 2. Transplant configuration for *Halodule wrightii* restoration sites in West Bay. Transplanting unit (TPU) densities are represented by shading: black = 0.25 m centers, dark stippling = 0.5 m centers, light stippling = 1.0 m centers, white = control. Each strip is 8 m wide, 54 m long, and oriented perpendicular to the shoreline.

ters (408 TPUs); and 12 rows at 33 TPUs per row on 0.25 m centers (396 TPUs; Fig. 2). Thus, 3492 TPUs were placed into each bed and 10,476 TPUs were placed overall. A fourth 8 m × 54 m strip within each bed was left bare as a control to monitor natural recruitment (Fig. 2). Maximum and minimum water depths on the seaward edges of the beds during spring tides were 100–110 cm and 30–40 cm. Water depths of 0–10 cm were observed after passage of winter cold fronts. On any given sampling date, actual water depth differences were 15–20 cm between shallow and deep edges at Redfish and Snake Island Coves. Similar variations in water depth were noted along transects in the natural seagrass bed in Christmas Bay.

Monitoring Transplant Success

Fonseca (1994) suggested several criteria for judging transplant success relative to that of natural beds, of which we compared: (1) survival and propagation (spread from the transplanted material) of TPUs at each restoration site, (2) seagrass coverage, (3) densities of new shoots adjacent to randomly selected TPUs, and (4) annual assessment of aboveground and belowground

biomass. We also examined the aboveground to belowground biomass ratio (root-to-shoot ratio or RSR) because it serves as an indicator of seagrass carbon storage capacity that is susceptible to stress (Lee & Dunton 1997). We added a one-time comparison of leaf chlorophyll, root-rhizome carbohydrate-carbon, and leaf and root-rhizome total carbon, hydrogen, and nitrogen (CHN) contents as measures of seagrass health because they adapt in response to local stresses such as reduced light regimes (Pulich 1989; Lee & Dunton 1997). Transplanted and natural grassbeds were examined monthly during June–October 1994 and in April, June, and September 1995. A limited assessment of biomass was conducted in June 1996, as was an estimate of the total areal extent of transplant beds both from this study and from experimental plantings in August 1993 (Hammerstrom et al. 1998).

Halodule survival and propagation were determined by snorkeling over randomly chosen rows of TPUs at each site. Rows were located by counting TPUs along the edges of each section and then following each row across that section. Rows were 8 m long and contained 9, 17, or 33 TPUs, depending on planting density. TPUs were scored as live (presence of green leaves) or dead (no leaves apparent), and live TPUs were scored as propagating (rhizomes extending beyond the original TPU) or dormant (no rhizomes outside the original TPU). In June 1994 we examined two separate sets of 350 TPUs from randomly selected rows at each site (approximately 10% of the TPUs in each density-depth combination) to determine whether future assessments of survival and spread would require checking more or less than 20% of the TPUs. There was no difference in TPUs alive (95.4% each) and propagating (85.7% versus 86.9%) between the two sets of 350 TPUs. In all subsequent months, therefore, we examined a total of 360 TPUs (40 TPUs from each of the nine density-depth combinations) in randomly selected rows at each restoration site. In addition, we swam transects across control sections to determine whether these areas remained nonvegetated.

In June 1996 we surveyed the entire fenced area of Redfish Cove to estimate the areal coverage of *Halodule*. We marked the edges of each patch with poles and snorkeled across each patch to make sure coverage was homogeneous. We then measured the lengths and widths of rectangular patches or diameters of circular patches and summed these areas to estimate total area covered.

The sampling design for monitoring *Halodule* shoot density and biomass was derived from a series of 25 samples collected with a core 10 cm in diameter taken to a depth of 5 cm in Christmas Bay during October 1993. The data set was subjected to a power analysis after $\log(x + 1)$ transformation (Sokal & Rohlf 1981),

which indicated that nine samples (3 *Halodule* densities × 3 water depths) would permit detection of a 100% difference in mean shoot density or biomass between any two beds with $\alpha = 0.10$ and $1 - \beta = $ power $= 0.90$. This sample size ($n = 9$) was used to monitor coverage, new shoot density, and above- and belowground biomass after transplanting. *Halodule* coverage was determined initially by placing a small quadrat (37.5 cm × 37.5 cm, divided into 25 squares, each 7.5 cm × 7.5 cm; TPUs were 7.5 cm in diameter) over nine randomly selected TPUs at each restoration site or at nine random points along each Christmas Bay transect and then counting the squares with shoots inside. We added a larger quadrat in October 1994 after coalescence of TPUs in the high-density areas, indicating that *Halodule* had grown beyond the edges of the small quadrat. The large quadrat (1 m × 1 m, divided into 25 squares, each 20 cm × 20 cm) was placed immediately adjacent to a randomly selected side of the small quadrat. Because the small quadrat was placed directly over a TPU, densities of new shoots resulting from propagation were counted in a randomly selected adjoining square (56.25 cm^2).

Aboveground and belowground biomass of transplanted and natural grassbeds was compared at the end of the first two growing seasons (October 1994, September 1995) and in the middle of the third season (June 1996). In the first two periods, two 10 cm-diameter, 10 cm-deep cores were pooled from each of the nine density-depth combinations at Redfish Cove and Snake Island East (1994 only) and from three randomly selected locations along each of the three transects in Christmas Bay. In June 1996, a single 10 cm-diameter, 10 cm-deep core was taken from each of nine density-depth combinations at Redfish Cove, from three randomly selected locations along each of the three transects in Christmas Bay, and from three random sites at the seaward edge of each Christmas Bay transect. Two cores were taken on the first two sampling dates due to sparse aboveground biomass at the transplant sites, whereas by the third sampling date the biomass was visibly higher. Cores were acceptable only if they excluded any original TPU material (identifiable by fragments of peat pots). Shoots were separated from roots and rhizomes (hereafter termed roots), and leaf epiphytes were scraped off. Each component was then dried at 60°C for 24 hours and weighed. Only *Halodule* was collected at the restoration sites. Christmas Bay samples were all *Halodule* in October 1994 but were mixtures of *Halodule*, *Halophila*, and *Ruppia* in September 1995 and June 1996.

Leaf chlorophyll, root carbohydrate-carbon, and leaf and root carbon, hydrogen, and nitrogen (CHN) contents of *Halodule* were compared among Redfish Cove, Snake Island East, and Christmas Bay in October 1994. Samples were collected as above for biomass, and

leaves were scraped to remove epiphytes. Ten to 12 leaves from each core were detached from shoots, then frozen in plastic bags. Remaining shoots and roots were separated, dried at 60°C to a constant weight, then stored in plastic bags. Total leaf chlorophyll (chl) and ratios of chl *a* to chl *b* were measured according to Dunton and Tomasko (1994). Carbohydrate-carbon of a subsample of dried roots was measured by the MBTH analysis method (Lee & Dunton 1997). CHN analyses were performed on separate subsamples of dried leaves and roots with a Carlo-Erba Model 1108 Elemental Analyzer. Comparisons were made among sites, rather than targeting criteria or standards, because chlorophyll, carbohydrates, and CHN vary seasonally as well as when plants are stressed (Pulich 1989; Lee & Dunton 1997).

Statistical Analyses

Habitat-related floral differences during each sampling period were examined by analysis of variance (ANOVA). We tested for differences in percent TPUs alive, percent TPUs propagating, percent seagrass coverage, densities of new shoots that had expanded beyond the original TPUs, and aboveground and belowground biomass. One-way ANOVA was used to compare differences in floral attributes among sites and the effects of fertilizer addition for each sampling period. Three-way ANOVA examined the effects of site, planting density, and water depth on floral components within restored areas only. Two-way ANOVA examined the effects of site and sediment depth on concentrations of sediment trace metals. All ANOVAs were conducted with balanced cell sizes, except in comparisons of the effects of fertilizer addition. Data were transformed prior to ANOVA as follows: arcsine for percentages; $\log(x + 1)$ for seagrass shoot counts and biomass; and $\sqrt{(x + 0.5)}$ for sediment iron (Sokal & Rohlf 1981). Multiple comparison of treatment means employed Ryan's Q test for balanced designs or the GT2 test for unbalanced designs (Day & Quinn 1989). All analyses were conducted with SAS personal computer software (SAS Institute, Inc. 1985).

Results

Suitability of Restoration Sites

Temperature and salinity appeared similar among sites. A typical growing season temperature cycle was noted, with highest temperatures during July–September and the passage of two cold fronts in October (Fig. 3). Continuous data sonde readings indicated that mean temperatures were within 1°C between Snake Island Cove and Christmas Bay in 1994 and were highly correlated (adjusted $R^2 = 0.918$, $p < 0.001$). Temperatures again were similar among sites during discrete site visits in
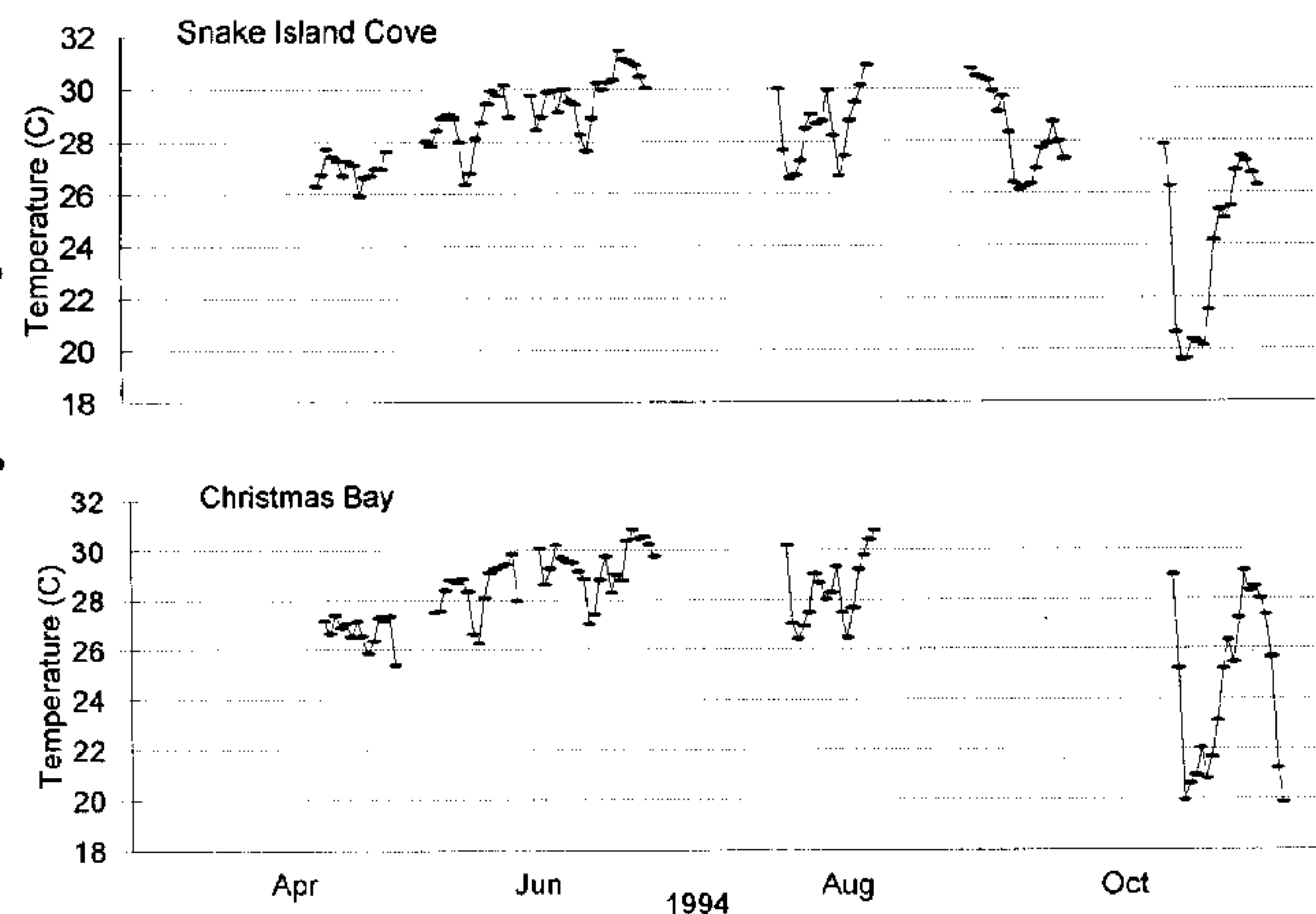
Figure 3. Daily mean water temperature of restored (Snake Island Cove) and natural (Christmas Bay) seagrass beds during the period 20 April–3 November 1994.

Figure 5. Daily total incident and submerged photon flux densities (mol/m²/day) in Snake Island Cove.

1995 and averaged 21°C in April, 29°C in June, and 28°C in September. Salinities were usually within 5‰ between sites during 1994 and were highly correlated (adjusted $R^2 = 0.719$, $p < 0.001$; Fig. 4). Salinities generally remained between 20‰ and 35‰, except for brief appearances of fresh water caused by locally heavy rainfall that depressed salinities for 5–7 days (Fig. 4). Salinities observed during site visits in 1995 were similar among sites and averaged 20‰ in April, 18‰ in June, and 27‰ in September.

Incident photon flux densities (PFD) were typically between 30 and 60 mol/m^{-2}/day at Snake Island Cove (mean = 36 mol/m^{-2}/day) and between 30 and 75 mol/m^{-2}/day at Christmas Bay (mean = 51 mol/m^{-2}/day; Figs. 5 and 6). Submerged PFD values were usually between 10 and 30 mol/m^{-2}/day at both Snake Is-

land Cove (mean = 19 mol/m^{-2}/day) and Christmas Bay (mean = 21 mol/m^{-2}/day). Total submerged quanta received over the periods of record were 2627 mol/m^{-2} over 129 days at Snake Island Cove and 3207 mol/m^{-2} over 147 days at Christmas Bay. Light transmittance to the bottom typically exceeded 40% in Snake Island Cove (daily mean = 54% ± 3% SE) and was somewhat higher in the spring than during summer and fall (Fig. 7). But light transmittance in Christmas Bay (daily mean = 44% ± 3%) exhibited a seasonal decline from about 60% in spring to 30% in fall. Light transmittance in Redfish Cove was not monitored but likely was higher than in Snake Island Cove, because Redfish Cove is about 10 cm shallower. These daily and total underwater light measurements were similar to those recorded by Dunton (1994) for a healthy *Halodule*
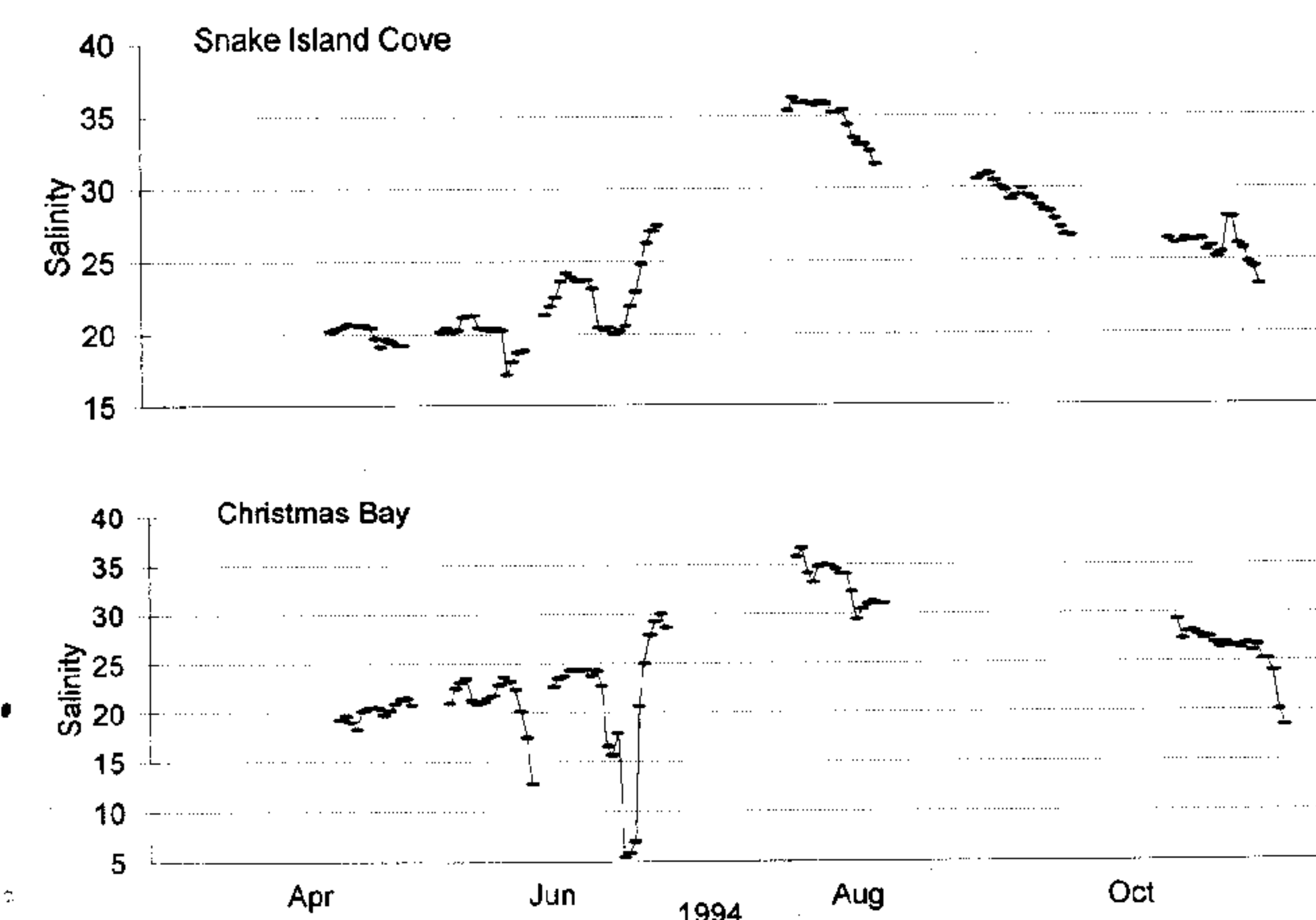

Figure 4. Daily mean salinity of restored (Snake Island Cove) and natural (Christmas Bay) seagrass beds during the period 20 April–3 November 1994.
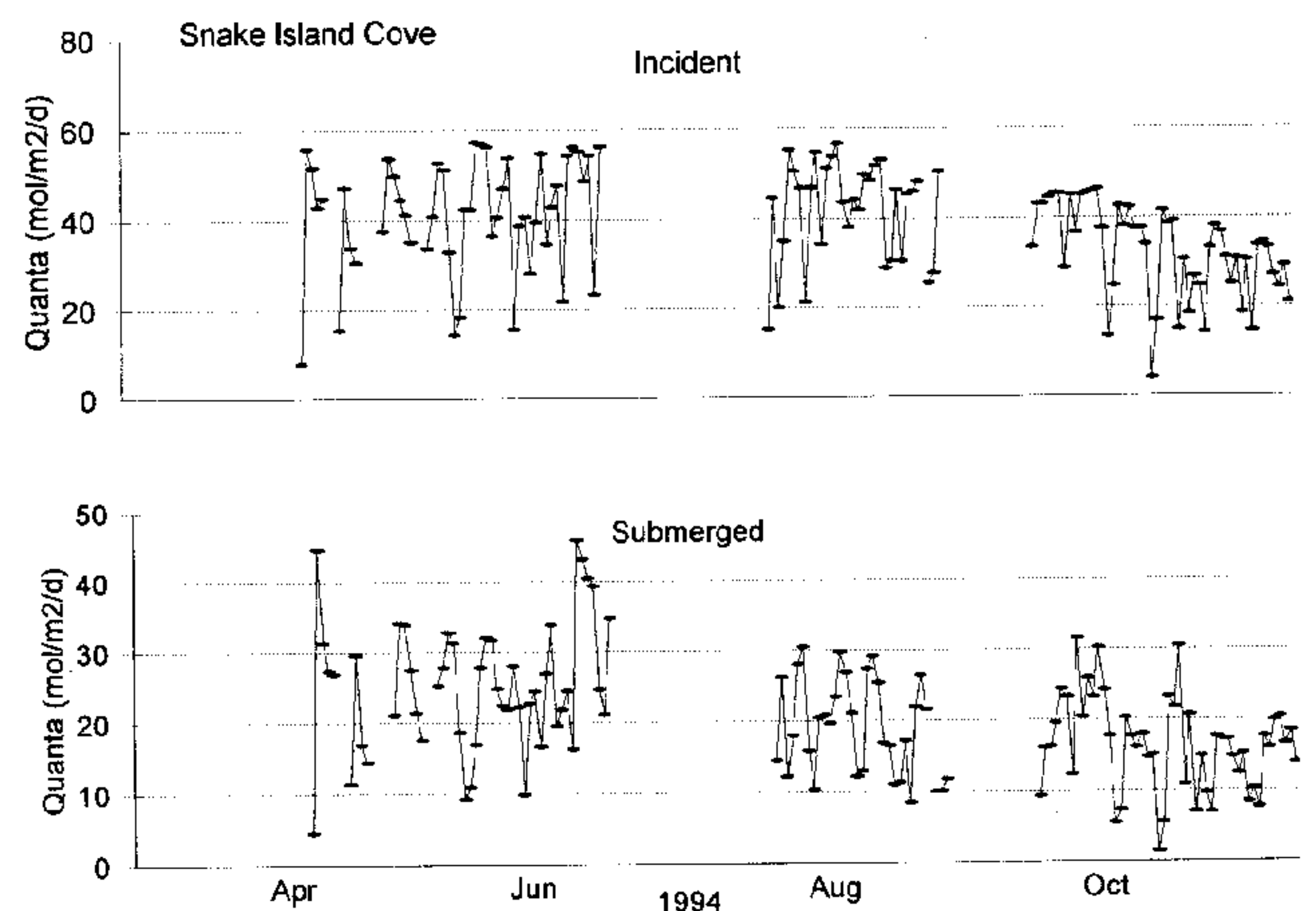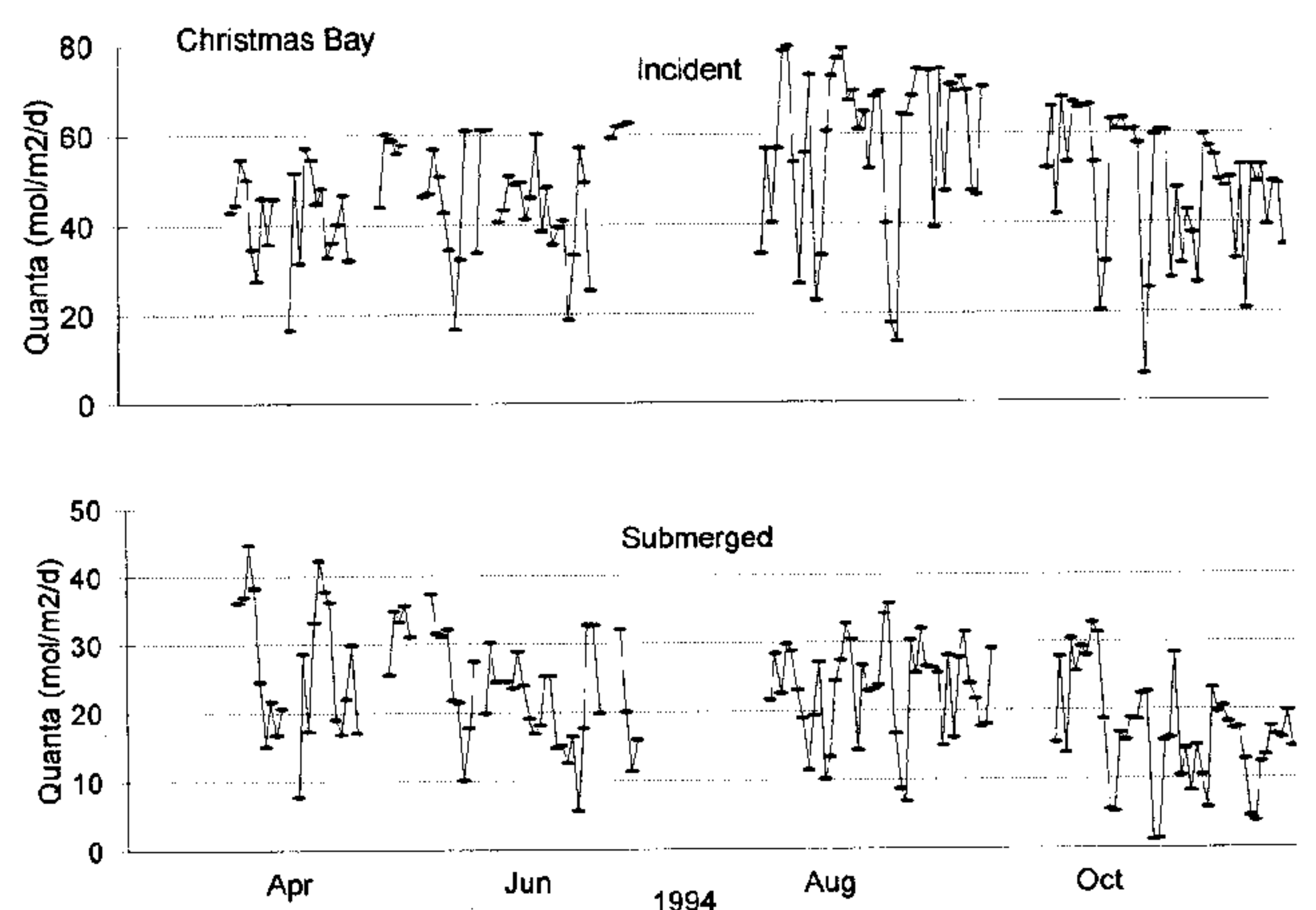

Figure 6. Daily total incident and submerged photon flux densities (mol/m^{-2}/day) in Christmas Bay.
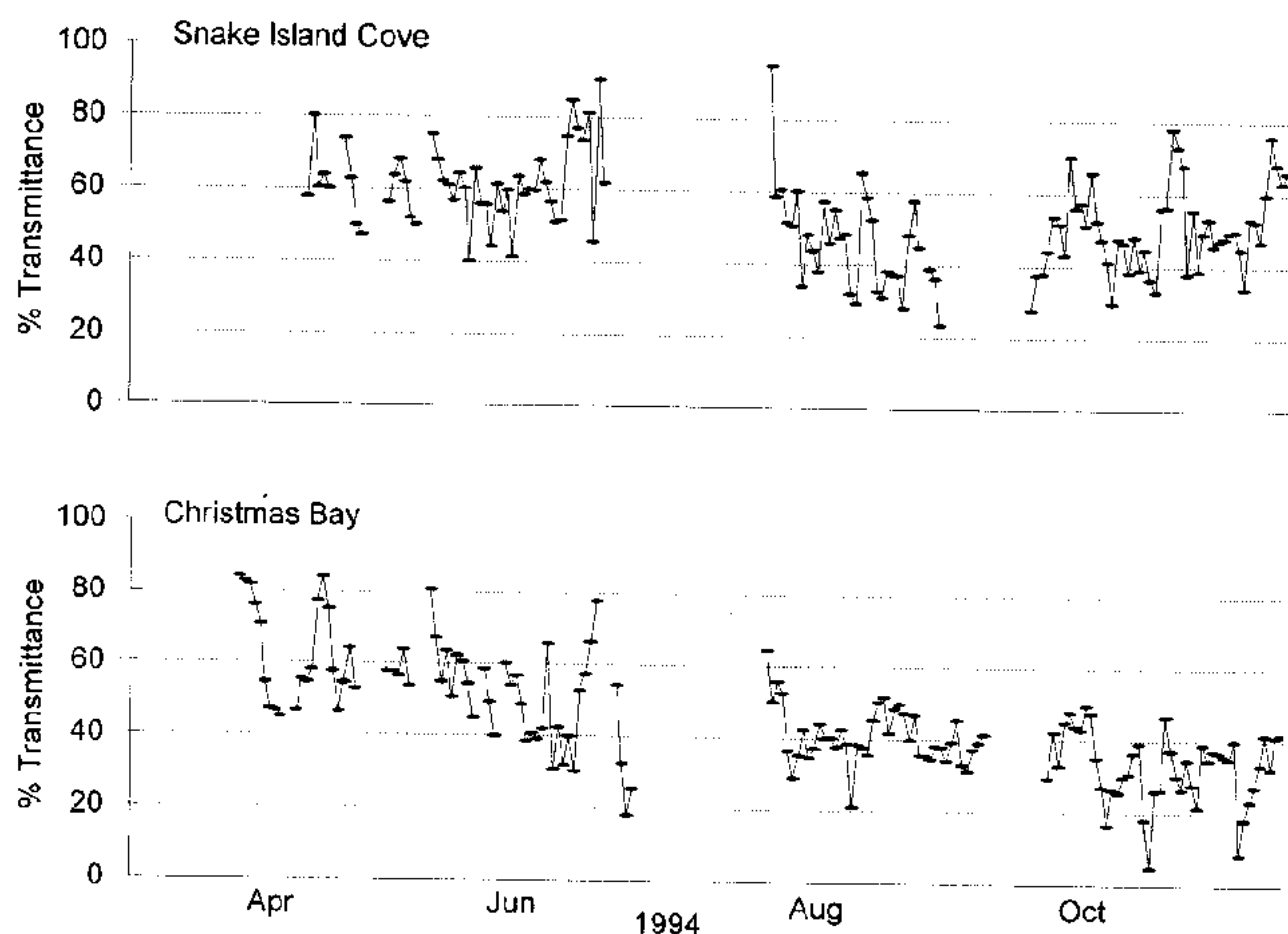
Figure 7. Daily variation in transmittance of incident light (%) to the bottom in Snake Island Cove and Christmas Bay.

bed in the East Flats area of Corpus Christi Bay, Texas, where submerged PFD remained consistently above 10 $mol/m^{-2}/day$ for five years. These values also exceeded the 18% surface irradiance threshold suggested by Dunton (1994) for light stress of *Halodule*.

Differences in light transmittance among sites were likely due to higher turbidities in Christmas Bay than in Snake Island Cove or Redfish Cove. Although continuous records of turbidity were not available, periodic checks during site visits in 1994 revealed that turbidity was higher in Christmas Bay (26 NTU ± 5.8 SE) than in either Redfish Cove (13 NTU ± 1.7) or Snake Island Cove (10 NTU ± 1.8). Turbidity appeared to be lower in 1995 (Christmas Bay, 11 NTU ± 2.0; Redfish Cove, 4 NTU ± 0.7; Snake Island Cove, 7 NTU ± 1.6).

All sediment trace metals except lead exhibited significant differences among some means for the three site-depth combinations (Table 1). Iron, copper, and nickel were significantly higher in Christmas Bay surface (0–5 cm) samples than in all other site-depth combinations. Concentrations of zinc and cadmium were higher in Christmas Bay surface samples than in all others except Redfish Cove. The only significant differences in manganese and arsenic were between Christmas Bay surface samples and Snake Island Cove surface and deep (6–10 cm) samples. There were few significant differences between Redfish Cove and Snake Island Cove sediments. Slightly lower concentrations of iron, manganese, and zinc were noted in Snake Island Cove deep samples and Redfish Cove shallow and deep samples than in Christmas Bay. In general, Snake Island Cove had lower sediment trace metal concentrations than did Christmas Bay (but not compared to Redfish Cove). Metal concentrations were up to an order of magnitude lower at the restoration sites and in Christmas Bay than in other nonvegetated parts of West Bay (Carr 1993) and were similar to concentrations observed in Corpus Christi Bay *Halodule* beds (Pulich 1980). Plotting concentrations of nonferrous sediment metals against iron levels generally revealed direct relationships. The three study sites clustered according to distinctive patterns, as observed for manganese and zinc versus iron (Fig. 8). High correlation coefficients were calculated, suggesting that sediment geochemical processes involving these metals were similar and were typical among sites.

Halodule Restoration

Halodule survival was significantly related to site, water depth, and planting density after the first three months of monitoring (Fig. 9). No significant ANOVA two- or three-way interactions were noted, except for a three-way interaction in September 1995 (due to mortality of

Table 1. Mean sediment trace metal concentrations ($\mu g/g$ dry sediment) and water content (%) in surface (0–5 cm) and subsurface (6–10 cm) layers at natural and restored seagrass sites.*

| | Christmas Bay | | Redfish Cove | | Snake Island Cove | | ANOVA Significance Levels | | | Other Sites | |
	0–5 cm	6–10 cm	0–5 cm	6–10 cm	0–5 cm	6–10 cm	Site	Depth	S × D	WB	CCB
Fe	4033	2267	2253	2530	1421	1363	<0.001	0.125	0.036	17000	3025
Mn	37.7	21.1	32.9	32.0	17.3	11.2	0.002	0.049	0.249	485	38.6
Zn	12.1	7.9	7.6	8.7	4.3	4.4	<0.001	0.251	0.043	64.8	40.7
Cu	2.41	0.98	0.93	1.25	0.79	0.82	0.001	0.039	<0.001	11.6	4.0
Ni	6.0	3.7	3.3	3.6	1.8	1.9	<0.001	0.158	0.052	15.2	7.1
Pb	4.6	3.1	4.3	4.2	4.4	4.0	NS	NS	NS	30.6	—
Cd	0.17	0.09	0.13	0.12	0.09	0.08	0.001	0.003	0.015	0.26	0.43
As	2.1	1.5	1.5	1.9	1.3	1.1	0.003	0.315	0.028	4.2	—
H2O	24.9	21.0	19.0	19.3	18.3	15.9	<0.001	0.068	0.266	—	—

*N = 4 per depth per site. Two-way analysis of variance (ANOVA) df = 2 (site), 1 (depth), and 2 (site × depth interaction). Comparative data are provided from five nonvegetated sites in West Bay (WB; Carr 1993) and four seagrass beds in the Corpus Christi Bay area (CCB; Pulich 1980).
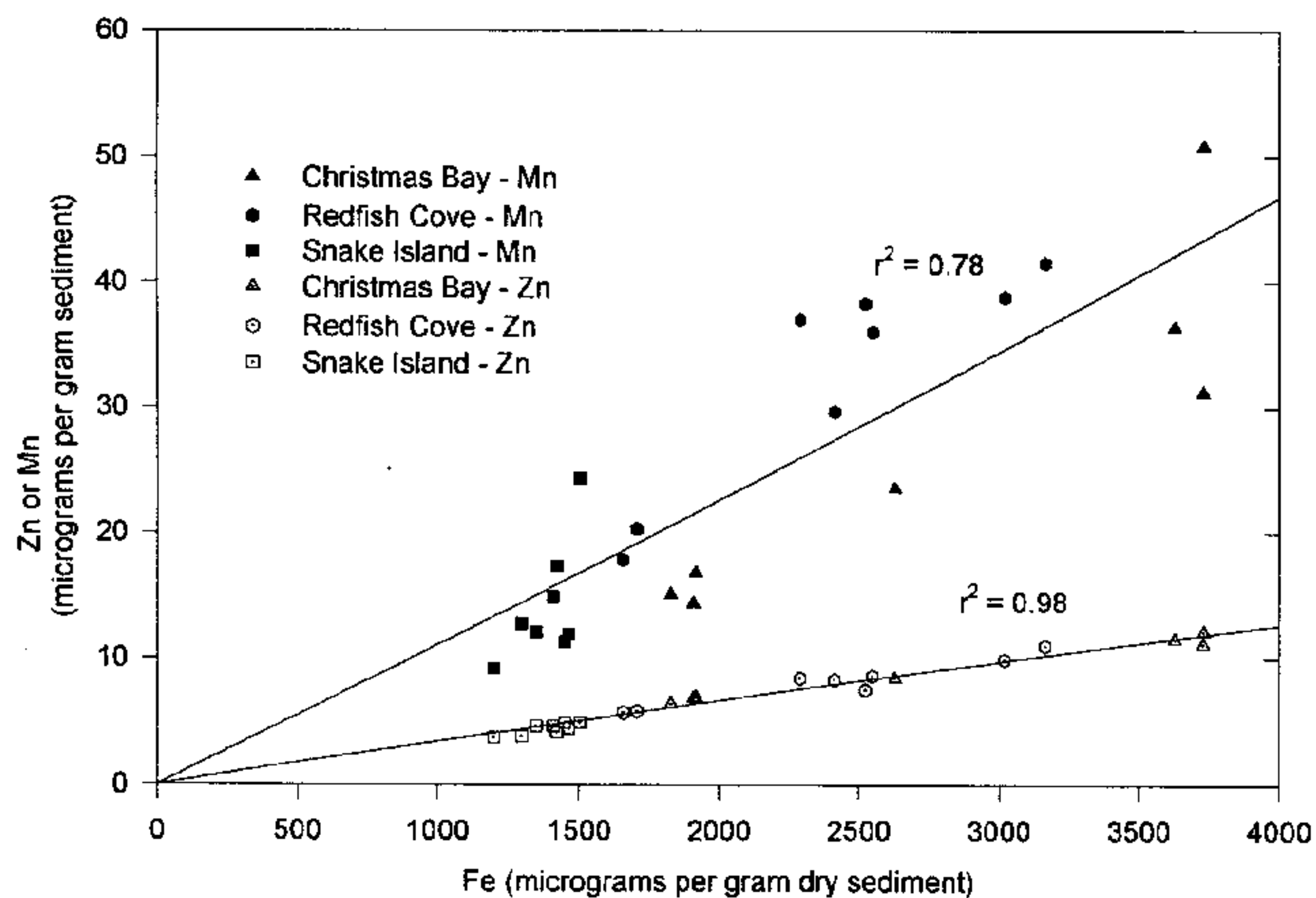
Figure 8. Relationships between iron and zinc or manganese concentrations (μg/g dry weight) in sediments at restoration sites (Snake Island, Redfish Cove) and in naturally occurring seagrasses (Christmas Bay). Data are combined over two sediment depth ranges (0–5 cm and 6–10 cm). $n = 4$ per depth range per site.

Snake Island Cove beds). Survival was significantly higher at Redfish Cove in late 1994 and in 1995 than at either Snake Island bed (Fig. 9a). Survival at Redfish Cove was 70% by September 1995, while almost all transplants at Snake Island Cove had died by then. The deterioration at Snake Island was a two-stage process that began at the end of the growing season in 1994, when average survival declined from 84% in August to 56% in September and October. Survival stabilized at

that level through June 1995, but then the decline increased through September 1995, resulting in a complete loss at Snake Island East and only 6% survival at Snake Island West (Fig. 9a). Qualitative transects across the length and width of each bed in June 1996 indicated no aboveground or belowground biomass in Snake Island East and only a small live *Halodule* patch (<1 m²) at the shallow edge of Snake Island West.

Halodule survival was also influenced by TPU planting density and water depth. In September and October 1994 and in April 1995, survival of high-density transplants was significantly greater than that of medium- and low-density plantings (Fig. 9b). In July and September 1995, survival of both high- and medium-density plantings significantly exceeded that of low-density beds. Survival in shallow sections often significantly exceeded that in middle depths and was always significantly greater than in deep sections (Fig. 9c). No significant differences in survival were attributable to the presence or absence of fertilizer (Fig. 9d). All nonfertilized transplants were located in Snake Island West and had died by September 1995.

Propagation of surviving *Halodule* plugs was significantly related to site, water depth, and planting density only during one or more of the first five months of monitoring (Fig. 10). No ANOVA two- or three-way interactions were significant, except for one three-way interaction in September 1995. Propagation was initially slower at Snake Island West than at Redfish Cove or Snake Island East, and significant differences were noted during August–October 1994 (Fig. 10a). Despite this, more than 75% of all TPUs showed signs of rhi-
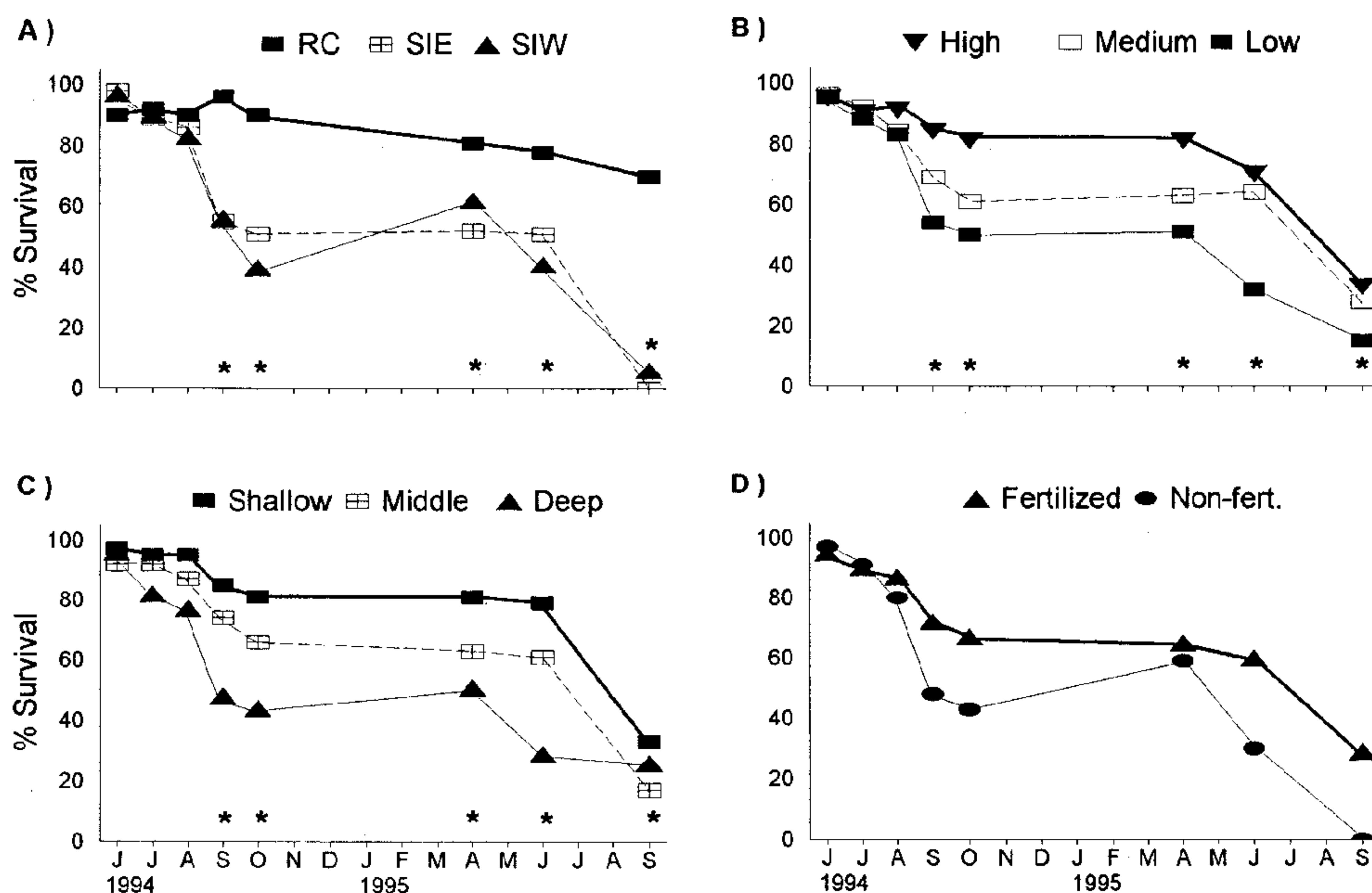

Figure 9. Mean seagrass survival (%) after transplanting in April–May 1994 as affected by site, where RC = Redfish Cove, SIE = Snake Island East, and SIW = Snake Island West (a); density, where high = 0.25-m centers, medium = 0.5-m centers, and low = 1.0-m centers (b); relative depth (c); and use of fertilizer (d). For a–c, $n = 360$ per category, except $n = 700$ in June 1994. For d, $n = 960$ fertilized and 120 nonfertilized, except $n = 1867$ and 233, respectively, in June 1994. Asterisk indicates significant differences (ANOVA, Ryan's Q (a–c) or GT2 (d), $p < 0.05$).
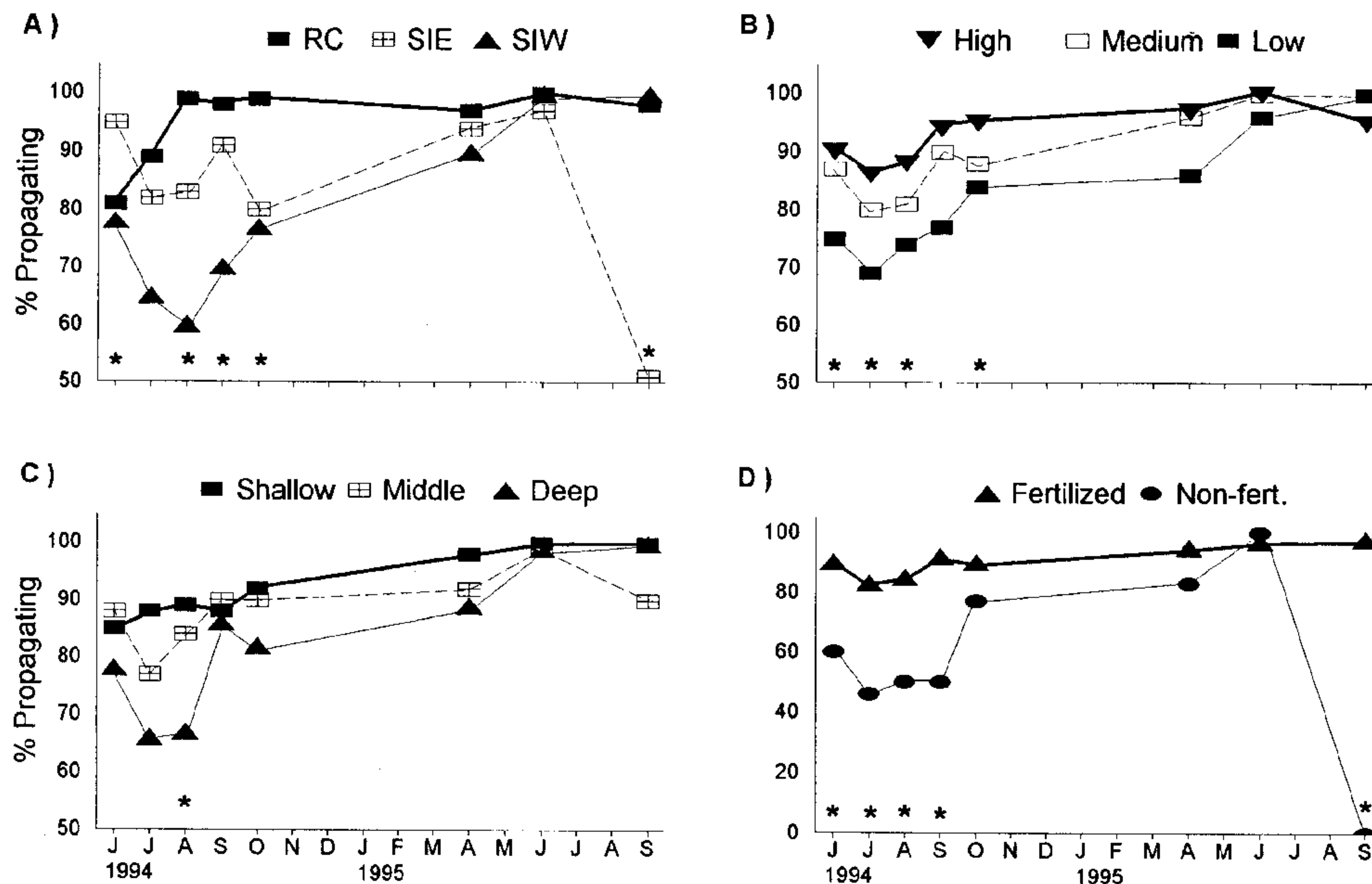
Figure 10. Mean propagation of surviving plugs (%) after transplanting in April–May 1994 as affected by site, where RC = Redfish Cove, SIE = Snake Island East, and SIW = Snake Island West (a); density, where high = 0.25-m centers, medium = 0.5-m centers, and low = 1.0-m centers (b); relative depth (c); and use of fertilizer (d). For a–c, $n = 360$ per category, except $n = 700$ in June 1994. For d, $n = 960$ fertilized and 120 nonfertilized, except $n = 1867$ and 233, respectively, in June 1994. Asterisk indicates significant differences (ANOVA, Ryan's Q (a–c) or GT2 (d), $p < 0.05$).

zome propagation outside the peat pots by the end of the first growing season. No differences in propagation were noted in 1995 except those due to complete mortality at Snake Island East.

The propagation of surviving TPUs was also influenced by planting density and water depth, but only during 1994. Expansion by high- or medium-density TPUs or both was significantly greater than by low-density TPUs in June, July, August and October 1994 (Fig. 10b). Propagation was similar among all water depths except in August 1994, when deep sections were significantly slower than elsewhere (Fig. 10c). Significantly higher incidences of expansion were also attributable to fertilizer additions during the first growing season (Fig. 10d). No recruitment of *Halodule* into the control areas at each site was observed at any time.

Halodule coverage was significantly related to site, water depth, planting density, and fertilizer addition during all six months of sampling (Fig. 11). No ANOVA two-way interactions were significant, but three-way
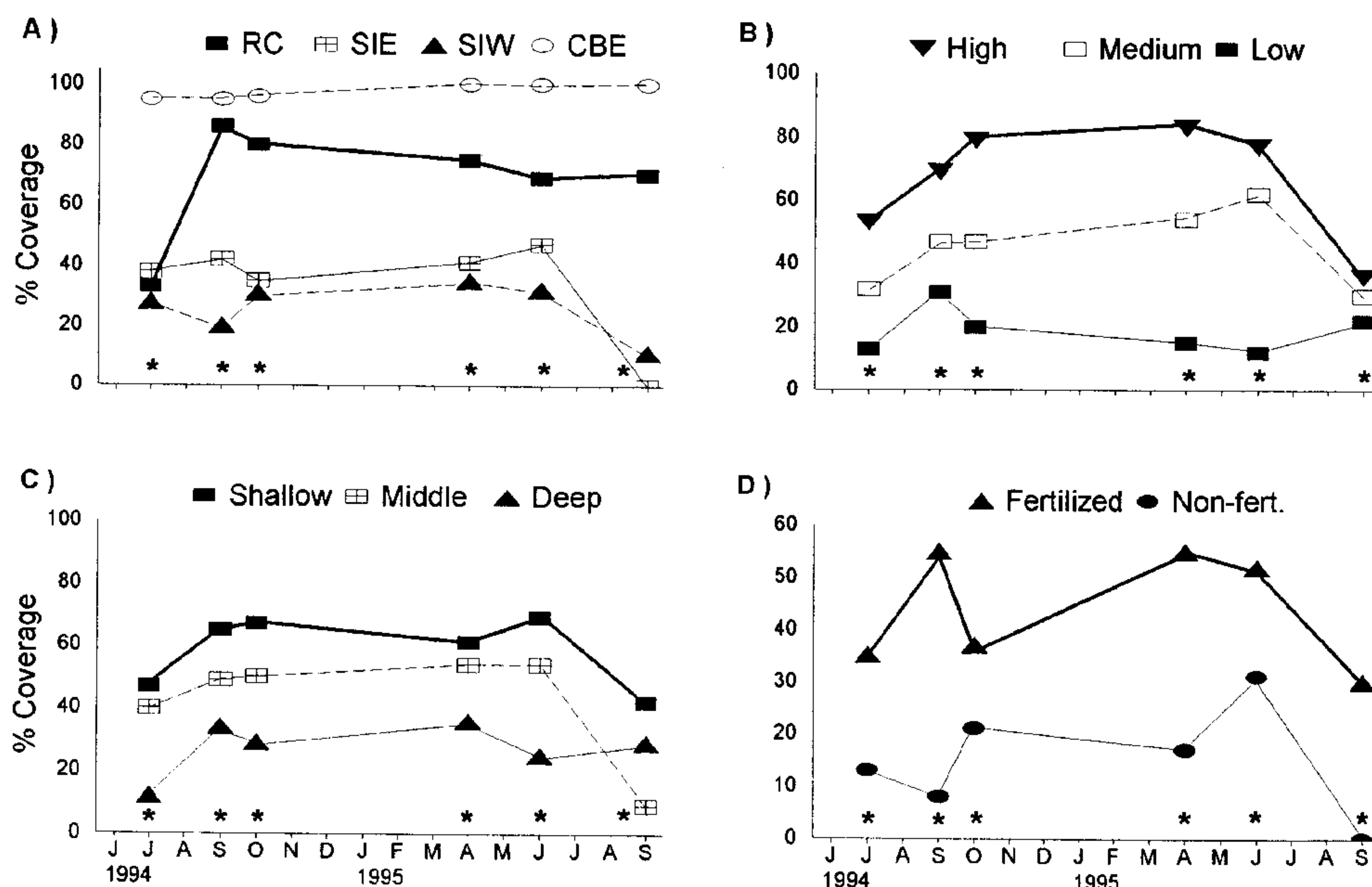

Figure 11. Mean seagrass coverage (%) after transplanting in April–May 1994 as affected by site, where RC = Redfish Cove, SIE = Snake Island East, SIW = Snake Island West, and CBE = Christmas Bay East (a); density, where high = 0.25-m centers, medium = 0.5-m centers, and low = 1.0-m centers (b); relative depth (c); and use of fertilizer (d). For a–c, $n = 360$ per category; for d, $n = 960$ fertilized and 120 nonfertilized. Asterisk indicates significant differences (ANOVA, Ryan's Q (a–c) or GT2 (d), $p < 0.05$).

interactions (site × depth × density) were significant in April, June, and September 1995, related to the mortality at Snake Island. There were no significant differences in seagrass coverage among the three Christmas Bay transects. Christmas Bay East was chosen as the index for comparison with restored beds because seagrasses were primarily *Halodule* (Christmas Bay West 1 and West 2 contained more *Halophila* and *Ruppia*, particularly in 1995). Coverage in restored beds was significantly lower than that of Christmas Bay East, except in September 1994 when Redfish Cove coverage equalled that of the natural bed (Fig. 11a). Coverage was significantly higher at Redfish Cove than at either Snake Island bed in all months except June 1994. Seagrass coverage always exhibited the relationship to planting density of high > medium > low (Fig. 11b), and these differences were significant with exceptions in July 1994 (medium = low) and August 1994 (high = medium). Significant differences in seagrass coverage were always found in relation to water depth, exhibiting the relationship shallow > middle > deep, except in September 1995 when shallow > deep > middle (Fig. 11c). In addition, fertilized plots always had significantly higher coverage than did nonfertilized plots (Fig. 11d).

Production of new shoots (i.e., those located outside the original TPUs) was significantly related to site, planting density, and water depth (Fig. 12). No ANOVA two- or three-way interactions were significant. New shoot densities (number per unit area) at the transplant sites were significantly lower than those in natural seagrass beds, except in September 1994 when Redfish Cove densities equalled those at Christmas Bay East (Fig.

12a). New-shoot densities at Redfish Cove were also significantly higher than those at Snake Island East and West in September 1994 and in June and September 1995. High-density plantings produced significantly higher numbers of new shoots than did medium- and low-density plantings in all but the first and last months (Fig. 12b). Shallow plantings produced higher new-shoot densities than did medium and deep plantings, significantly so in September and October 1994 and June 1995 (Fig. 12c). New-shoot densities in fertilized plots were significantly higher than in unfertilized plots in all but September 1995 (Fig. 12d).

Aboveground biomass was significantly lower in restored beds than in Christmas Bay during all times examined (Table 2), but the sparse leaf production at Redfish Cove was similar to that found on the seaward edge of the Christmas Bay bed in June 1996. Belowground biomass was significantly lower at Snake Island East than Christmas Bay in October 1994, and plants at Snake Island East had completely died out by September 1995. Redfish Cove root and rhizome biomass was similar to that of Christmas Bay in all years. The root-to-shoot ratios (RSR) of Christmas Bay and Snake Island Cove seagrasses were similar in October 1994, but relatively low aboveground biomass led to higher RSRs at Redfish Cove than at Christmas Bay. The apparent decline in root-rhizome biomass in Christmas Bay between 1994 and 1995 (Table 2) was likely due to the fact that we sampled *Halodule* almost exclusively in October 1994 but encountered mixtures of *Halodule*, *Halophila*, and *Ruppia* in September 1995 and June 1996. The latter two genera have lower belowground biomasses than
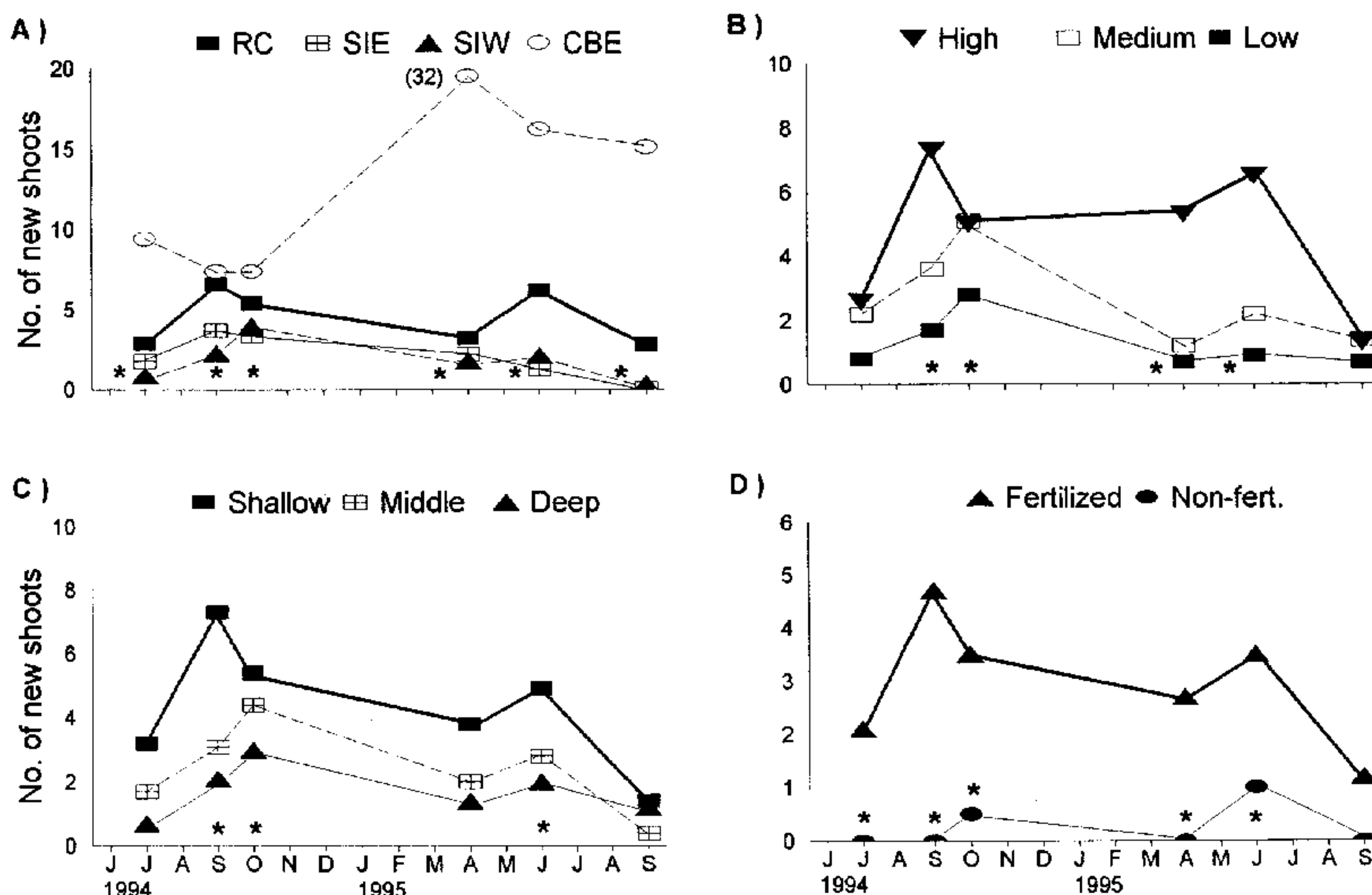

Figure 12. Mean new shoot densities (number per 56.25 cm^2) after transplanting in April–May 1994 as affected by site, where RC = Redfish Cove, SIE = Snake Island East, SIW = Snake Island West, and CBE = Christmas Bay East (a); density, where high = 0.25-m centers, medium = 0.5-m centers, and low = 1.0-m centers (b); relative depth (c); and use of fertilizer (d). For a–c, $n = 9$ per category; for d, $n = 24$ fertilized and 3 nonfertilized. Asterisk indicates significant differences (ANOVA, Ryan's Q (a–c) or GT2 (d), $p < 0.05$).

Table 2. Mean shoot and root-rhizome biomasses (dry g/m^2) and root-shoot ratios (RSR) of Christmas Bay transect (CBT) or seaward edge (CBS) and transplanted Redfish Cove (RFC) or Snake Island East (SIE) seagrasses.*

| Date | Element | Sites | | | | ANOVA | | |
		CBT	CBS	RFC	SIE	df	F	p
10/94	Shoots	18.99 a	—	4.63 b	5.14 b	2, 24	15.6	<0.001
	Roots	119.42 a	—	69.59 ab	41.58 b	2, 24	6.13	0.007
	RSR	7.08 b	—	15.26 a	8.58 b	2, 24	9.41	<0.001
9/95	Shoots	13.16 a	—	1.37 b	—	1, 16	45.76	<0.001
	Roots	23.86	—	23.27	—	1, 16	0.39	0.543
	RSR	2.27 b	—	19.35 a	—	1, 16	70.25	<0.001
6/96	Shoots	27.40 a	12.43 b	12.42 b	—	2, 24	5.29	0.012
	Roots	37.21 ab	17.18 b	67.66 a	—	2, 24	4.15	0.028
	RSR	1.45 b	1.50 b	4.60 a	—	2, 24	14.42	<0.001

*RFC and SIE were planted in April-May 1994. Analysis of variance (ANOVA) conducted on log(x + 1)-transformed data. Means indicated with differing letters are significantly different (n = 9, Ryan's Q test, p < 0.05). Dash = not tested.

Halodule (Pulich 1985; Fonseca 1989). Epiphyte loads were not quantified, but qualitative observations indicated minimal overgrowth at the restoration sites relative to that on leaves in Christmas Bay. No attached or drift macroalgae were observed in any of the seagrass beds or collected in any biomass samples. Large-scale bioturbation, such as excavation of pits by rays or crabs, was not observed.

An estimate of the health of transplanted versus naturally occurring *Halodule* was obtained in October 1994 by examining and comparing various indicators (Table 3). There were no significant differences in total leaf chlorophyll or in the ratio of chl *a* to chl *b* among sites. However, Christmas Bay leaf carbon, hydrogen, and nitrogen were significantly lower than those at both Redfish Cove and Snake Island East, which may be related either to growth metabolism differences or to inherent variation between populations in Christmas Bay grassbeds and Matagorda Bay donor beds. Belowground components at Snake Island Cove were exhibiting stress after the first growing season: root carbohydrate-

carbon and total carbon and hydrogen were significantly lower at Snake Island East than at either Redfish Cove or Christmas Bay (Table 3).

We estimated that the total area covered by *Halodule* at Redfish Cove, including surviving TPUs, was 1014 m^2 as of June 1996. *Halodule* coverage ranged from solid to patchy, with isolated patches as small as 2.3 m^2. Aboveground production appeared uniformly short and sparse, as reflected in the biomass samples (Table 2). The shallow edge had coalesced across intervening walkways and outward beyond the edges of the fence to form the largest continuous feature (370 m^2). We also located two 1 m^2 patches west of the main site that have survived since August 1993 following experimental plantings (Hammerstrom et al. 1998).

Discussion

The two sites chosen for seagrass restoration experiments represent some of the last shoreline of western Galveston Island that historically supported seagrasses

Table 3. Total chlorophyll (mg/g dry weight), chlorophyll *a*: chlorophyll *b* (*a:b* ratio), carbohydrate carbon (mg/g dry weight), and carbon, hydrogen and nitrogen contents (% dry weight) of Christmas Bay (CB), Redfish Cove (RFC), and Snake Island East (SIE) *Halodule wrightii* in October 1994.*

| | | Sites | | | ANOVA | |
		CB	RFC	SIE	F	p
Leaves	Chlorophyll	12.43	11.99	10.90	2.06	0.150
	a:b ratio	2.71	2.79	2.78	0.95	0.400
	Carbon	34.91 b	37.18 a	36.68 a	32.40	<0.001
	Hydrogen	5.10 b	5.59 a	5.48 a	41.55	<0.001
	Nitrogen	2.49 b	2.76 a	2.83 a	9.31	0.001
Roots/Rhizomes	Carbohydrate	157.10 a	162.00 a	129.8 b	8.25	0.002
	Carbon	35.09 a	34.80 a	31.23 b	12.93	<0.001
	Hydrogen	5.23 a	5.38 a	4.72 b	14.04	<0.001
	Nitrogen	1.37 ab	1.20 b	1.46 a	4.17	0.030

*Analysis of variance (ANOVA) df = 2, 24. Means indicated with differing letters are significantly different (Ryan's Q test, p < 0.05).

and that currently remains undisturbed by canal housing developments (Hammerstrom et al. 1998). The western tip of Galveston Island remains undeveloped beyond cattle ranching, although housing communities are encroaching and a nonfunctioning sewage treatment plant has been built near Redfish Cove in anticipation of future needs. On-site monitoring indicated that environmental conditions in the vicinity of the restoration sites resemble those of healthy *Halodule* beds in adjoining Christmas Bay and elsewhere in Texas. Temperatures and salinities at Snake Island Cove, Redfish Cove, and Christmas Bay were well within the tolerance ranges of *Halodule* (4–35°C and 1–60‰; Simmons, 1957; McMillan & Moseley 1967; McMahan 1968). Light transmittance (54% at Snake Island Cove, 44% at Christmas Bay) was above the critical level of 18% postulated for *Halodule* survival (Dunton 1994). Moreover, turbidities were higher in the extant seagrass bed at Christmas Bay than in Snake Island Cove, particularly in the first growing season. In general, sediment metal concentrations were lower at the restoration sites than in Christmas Bay or non-vegetated sites elsewhere in West Bay (Carr 1993). Levels of total iron, manganese, zinc, and other metals measured at our sites appear typical of other nontoxic bay sediments along the Texas coast (Trefrey & Presley 1976; Pulich 1980, 1982). Local differences in these concentrations may reflect slight differences in sediment organic content or grain size, since adsorption of most metals varies directly with organic content and inversely with grain size (Trefrey & Presley 1976; Cross & Sunda 1978). Indeed, sediment organic content in Christmas Bay exceeded that of the three transplant sites in August 1994 (3.23% versus 1.06–1.42%, respectively; C. Henderson, Texas A&M University at Galveston, Galveston, Texas, personal communication).

Results to date indicate that only Redfish Cove will support viable *Halodule* beds for more than one growing season (experimental plots have survived into their fourth growing season). But *Halodule* coverage, shoot densities, and aboveground biomass at Redfish Cove remained significantly below those of natural seagrass beds in Christmas Bay two years after transplanting. In Tampa Bay, persistent cover equivalent to that of natural *Halodule* beds was noted 1.8 years after transplanting (Fonseca et al. 1996a). Relatively nutrient-poor sediments may lead to lower shoot biomass (Short 1987; Van Tussenbroek 1995). Propagation from the initial TPUs was enhanced by fertilizer additions, at least during the first growing season, which is a possible indicator of short-term nutrient depletion. The reduced leaf length and leaf biomass at Redfish Cove might also have been caused by herbivores, which have been observed elsewhere to reduce shoot lengths by 50% or more (Fonseca et al. 1987).

The failure of transplants at Snake Island Cove remains unexplained yet intriguing. Episodes of dramatically increased mortality were noted each September during the two growing seasons monitored, with no known cause, and the site did not truly fail until the last date examined. Several possibilities for why an apparently benign site such as Snake Island Cove would not support seagrasses remain unexplored. These should be addressed prior to further restoration attempts. Water column or sediment contamination by hydrocarbons or biocides were not tested, nor were water or sediment nutrient levels. Nutrients or contaminants could be contained in runoff or groundwater discharges from a canal housing development near Snake Island Cove. West Bay has relatively good water and sediment quality (Ward & Armstrong 1992), but there are areas along the mainland side of West Bay where sediment contamination by unidentified chemicals stresses benthic communities (Carr 1993). Sediment nutrient chemistry should also be examined with reference to possible nutrient limitation at both Snake Island and Redfish Coves, since addition of time-release fertilizer stimulated propagation of transplants relative to those not receiving fertilizer. Growth of *Halodule* in the Corpus Christi Bay area was shown to be stimulated by organic nutrient additions (especially organic nitrogen) to sediments but not by additions of inorganic nutrients (Pulich 1982, 1985, 1989). Similarly, sediment enrichment with inorganic nutrients had no effect on biomass, density, or leaf production in *Thalassia hemprichii* when ambient pore water nutrient concentrations were high (Erftemeijer et al. 1994). Finally, bioturbation (e.g., excavation by rays or crabs) cannot be ruled out. Small fishes and decapods common to West Bay do disturb transplants (Hammerstrom et al. 1998), and Fonseca et al. (1996b) noted a 47% loss of TPUs in Tampa Bay due to bioturbation.

In addition to site, *Halodule* restoration success was affected by transplant density. Transplants generally survived better and propagated faster when planted in high and medium densities (0.25 m and 0.5 m centers) rather than in low densities (1.0 m centers). The relationships between planting density and survival or propagation have not been addressed previously. Transplants of *Halodule* at 26 sites in the Florida Keys, all on 1.0 m centers, had a mean survival rate of 72.8% after 12–16 months but varied from 0 to 100% among sites (Lewis 1987). Transplants on 0.5 m and 1.0 m centers in some of the same areas exhibited 100% survival after 4–8 months (Fonseca et al. 1987). More recently, discussion of planting density has centered only on the observation that increased planting densities enhance coverage rates (Fonseca 1994; Fonseca et al. 1996a). But planting density should influence local current velocities and sediment dynamics. In natural seagrass beds,

coverage is inversely related to current velocity, and higher current speeds induce greater sediment mobility (Fonseca et al. 1983). Our low density plantings (1.0 m centers) possibly experienced higher current velocities and more sediment movement, increasing the likelihood for abrasion or burial, than did higher density plantings. The possibility of physical disturbance by currents at Redfish Cove or other sites should be examined, particularly since 1.0 m-center plantings would be a faster, more economical way to fill the large area proposed for restoration (567 ha; Galveston Bay National Estuary Program 1995) than would 0.25 m or 0.5 m-center plantings.

Restoration success was also affected by water depth. Survival, coverage, and production of new shoots were greater when the TPUs were planted in the shallow and middle sections of each bed than when they were planted in the deep sections. Interestingly, propagation was similar among all three depths after the first three months monitored. Even though we had only one set of meters available to monitor water quality and light at one restoration site, it was unlikely that gradients in bottom temperature or salinity were experienced over the 54 m length and 15–20 cm depth difference between the shallowest and deepest TPUs at either site. A gradient in light transmittance would be more critical and more likely. Our light meters were placed at mid-depth and indicated that submerged light was more than sufficient to support *Halodule* growth at that depth or shallower. We do not have a measure of the light transmission when depths were 7.5–10 cm greater than the depth of the meter. Dunton (1994) found annual variation in light transmittance of 9–22% on the outer edge of a shallow (≤0.6 m depth) *Halodule* bed on the Blackjack Peninsula of San Antonio Bay, Texas. Even though root-shoot ratios were similar to those of other sites in the Corpus Christi Bay area, total biomass at Blackjack Peninsula was an order of magnitude less than elsewhere, and plants disappeared from waters deeper than 0.6 m. Thus, a small change in depth can affect the survival and growth of established seagrass populations when they are living at or near tolerance limits, and intuitively it would seem more critical for transplants placed in such situations. Our results suggest that an array of light meters should be deployed to estimate maximum depth limits prior to future restorations at Redfish Cove or elsewhere.

Addition of time-release fertilizer to TPUs was found to enhance propagation and thus coverage and new-shoot densities, even though survival was not affected. Response to fertilizer additions elsewhere has been found to be positive or neutral (Fonseca et al. 1994). We agree with the recommendation that, for minimal cost and no reported adverse effects, fertilizer additions should be an integral part of seagrass restoration.

Assessment of certain physiological indices, particularly root carbohydrate content, may serve as early indicators of transplant stress. Even though root and shoot biomasses at Snake Island Cove were significantly lower than those of Christmas Bay at the end of the first growing season, the root-shoot ratios and chlorophyll contents were similar. The conclusion based on these data would have been that the bed was sparse but healthy. At the same time, however, root carbohydrate concentrations at Snake Island Cove were significantly below those of either Christmas Bay or Redfish Cove. This type of decline also was observed by Genot et al. (1994) in stressed *Posidonia oceanica* transplants. Carbohydrate storage over the growing season is critical for overwintering and the longer-term survival of seagrasses, and a depressed carbohydrate content is suggested as an indicator of stress (Lee & Dunton 1997). Among another indicators, assessment of CHN and chlorophyll contents may not be as useful. We did see significant depressions in root carbon and hydrogen at Snake Island Cove relative to those at Redfish Cove, but root nitrogen and leaf CHN were similar between these sights. Changes in carbon and hydrogen are not specific and could be related to changes in carbohydrate content, structural components such as cellulose, other cellular constituents, or a combination of these. Seasonal and monthly variations in shoot and root carbon and nitrogen contents in healthy seagrasses have been noted by Harrison and Mann (1975), Pulich (1982), and Pérez-Lloréns and Niell (1993), as have short-term variations in chlorophyll content (Dunton & Tomasko 1994). Both leaf nitrogen and RSR normally decrease in *Halodule* over the growing season as new plant tissues are produced (Pulich 1982; Dunton & Tomasko 1994). We would therefore recommend comparing root carbohydrate content in natural beds and in test beds planted at least one growing season before restoring any given site to its fullest extent.

There are several recommendations to be made concerning transplanting seagrasses into the Galveston Bay estuary, particularly because of the ambitious plan to restore 567 ha (1400 acres) of submerged aquatic vegetation (Galveston Bay National Estuary Program 1995). Given that we do not know much about the physical, chemical, and biological status of specific restoration sites, it would benefit resource agencies to conduct various screening tests (including the planting of small experimental beds) prior to full-scale restoration at any given site in West Bay. These tests should help avoid the catastrophic failure we found in Snake Island Cove. Reconnaissance of potential chemical contaminants such as biocides or hydrocarbons in sediments and water along the remaining undeveloped shoreline needs to be conducted, as do assessments of sediment nutrient chemistry with reference to possible nutrient limitation. Cur-

rent velocities should be examined during times of maximum tidal ranges, particularly if widely spaced plantings (e.g., 1.0 m centers) are anticipated. An array of light meters should be deployed at each site to estimate maximum depth limits for transplants. Once a potential planting site passes these screening criteria, test plantings on the order of 100 m² beds should be installed and monitored for at least two growing seasons. Monitoring programs should continue assessment of light regimes, annual fall assessments of root carbohydrate content, and periodic checks of survival, propagation, and production of new shoots. Our experiments and those of Fonseca et al. (1994, 1996a) and Hammerstrom et al. (1998) indicate that, once final sites are delimited, transplanting should be conducted during spring months on 0.25 m or 0.5 m centers, in relatively shallow water, by the peat pot method, and including time-release fertilizer. Restored beds should then be fenced with relatively small mesh screen (30 mm × 30 mm or smaller) for 60–90 days. These techniques have proven to be cost-effective and should foster the rapid growth and coalescence of seagrasses and thus long-term resistance to physical and biotic disturbances in West Bay. These experiments also indicate that monitoring of transplant success should be planned, at least annually in the fall, for three growing seasons.

Finally, we must point out that seagrass transplanting has a history of mixed success, even under the best of conditions (Lewis 1987). The goals of equal or greater persistent acreage with faunal equivalency to natural seagrass beds remain elusive after 50 years of experimentation with a "proven" technology (Fonseca 1992). The fact that restored beds at Snake Island Cove failed should not overshadow an apparently persistent, albeit sparse, new *Halodule* bed at Redfish Cove.

LITERATURE CITED

Adair, S. E., J. L. Moore, and C. P. Onuf. 1994. Distribution and status of submerged vegetation in estuaries of the upper Texas coast. Wetlands **14**:110–121.

Carr, R. S. 1993. Sediment quality assessment survey of the Galveston Bay system. GBNEP-30. Galveston Bay National Estuary Program, Houston, Texas.

Cross, F. A., and W. G. Sunda. 1978. Relationships between bioavailability of trace metals and geochemical processes in estuaries. Pages 429–442 in M. Wiley, editor. Estuarine interactions. Academic Press, New York.

Day, R. W., and G. P. Quinn. 1989. Comparison of treatments after an analysis of variance in ecology. Ecological Monographs **59**:433–463.

Duke, T. W., and W. L. Kruczynski. 1992. Status and trends of emergent and submerged vegetated habitats, Gulf of Mexico, U.S.A. EPA 800-R-92-003. Gulf of Mexico Program, U. S. Environmental Protection Agency, Stennis Space Center, Mississippi.

Dunton, K. H. 1994. Seasonal growth and biomass of the subtropical seagrass *Halodule wrightii* in relation to continuous measurements of underwater irradiance. Marine Biology **120**: 479–489.

Dunton, K. H., and D. A. Tomasko. 1994. *In situ* photosynthesis in the seagrass *Halodule wrightii* in a hypersaline subtropical lagoon. Marine Ecology Progress Series **107**:281–293.

Erftemeijer, P. L. A., J. Stapel, M. J. E. Smekens, and W. M. E. Drossaert. 1994. The limited effect of *in situ* phosphorus and nitrogen additions to seagrass beds on carbonate and terrigenous sediments in South Sulawesi, Indonesia. Journal of Experimental Marine Biology and Ecology **182**:123–140.

Fonseca, M. S. 1989. Sediment stabilization by *Halophila decipiens* in comparison to other seagrasses. Estuarine Coastal and Shelf Science **29**:501–507.

Fonseca, M. S. 1992. Restoring seagrass systems in the United States. Pages 79–110 in G. W. Thayer, editor. Restoring the nation's marine environment. Sea Grant Publication UM–SB–TS–92–06. University of Maryland, College Park, Maryland.

Fonseca, M. S. 1994. A guide to planting seagrasses in the Gulf of Mexico. Sea Grant Publication TAMU–SG–94–601. Texas A&M University, College Station, Texas.

Fonseca, M. S., J. C. Zieman, G. W. Thayer, and J. S. Fisher. 1983. The role of current velocity in structuring eelgrass (*Zostera marina*) meadows. Estuarine Coastal and Shelf Science **17**: 367–380.

Fonseca, M. S. W. J. Kenworthy, and G. W. Thayer. 1987. Transplanting of the seagrasses *Halodule wrightii, Syringodium filiforme*, and *Thalassia testudinum* for sediment stabilization and

habitat development in the southeast region of the United States. Technical report EL–87–8. U. S. Army Engineers Waterways Experiment Station, Vicksburg, Mississippi.

Fonseca, M. S., W. J. Kenworthy, F. X. Courtney, and M. O. Hall. 1994. Seagrass planting in the southeastern United States: methods for accelerated habitat development. Restoration Ecology 2:198–202.

Fonseca, M. S., W. J. Kenworthy, and F. X. Courtney. 1996a. Development of planted seagrass beds in Tampa Bay, Florida, USA. I. Plant components. Marine Ecology Progress Series 132:127–139.

Fonseca, M. S., D. L. Meyer, and M. O. Hall. 1996b. Development of planted seagrass beds in Tampa Bay, Florida, USA. II. Faunal components. Marine Ecology Progress Series 132:141–156.

Galveston Bay National Estuary Program. 1995. The Galveston Bay Plan. GBNEP-49. Galveston Bay National Estuary Program, Houston, Texas.

Genot, I., G. Caye, A. Meinesz, and M. Orlandini. 1994. Role of chlorophyll and carbohydrate contents in survival of *Posidonia oceanica* cuttings transplanted to different depths. Marine Biology 119:23–29.

Hammerstrom, K., P. Sheridan, and G. McMahan. 1998. Potential for seagrass restoration in Galveston Bay, Texas. Texas Journal of Science 50:35–50.

Harrison, P. G., and K. H. Mann. 1975. Chemical changes during the seasonal cycle of growth and decay in eelgrass (*Zostera marina*) on the Atlantic coast of Canada. Journal of the Fisheries Research Board of Canada 32:615–621.

Kantrud, H. A. 1991. Wigeongrass (*Ruppia maritima* L.): a literature review. Fish and Wildlife Research 10. U. S. Fish and Wildlife Service, Washington, D.C.

Lee, K.-S., and K. H. Dunton. 1997. Effects of *in situ* light reduction on the maintenance, growth and partitioning of carbon resources in *Thalassia testudinum* Banks ex König. Journal of Experimental Marine Biology and Ecology 210:53–73.

Lewis, R. R. 1987. The restoration and creation of seagrass meadows in the southeastern United States. Pages 153–173 in M. J. Durako, R. C. Phillips, and R. R. Lewis, editors. Proceedings of the Symposium on Subtropical-Tropical Seagrasses of the Southeastern United States. Marine Research Publication 42. Florida Department of Natural Resources, St. Petersburg, Florida.

McMahan, C. A. 1968. Biomass and salinity tolerance of shoalgrass and manateegrass in lower Laguna Madre, Texas. Journal of Wildlife Management 32:501–506.

McMillan, C., and F. N. Moseley. 1967. Salinity tolerances of five marine spermatophytes of Redfish Bay, Texas. Ecology 48:503–506.

Pérez-Lloréns, J. L., and F. X. Niell. 1993. Seasonal dynamics of biomass and nutrient content in the intertidal seagrass *Zostera noltii* Hornem. from Palmones River Estuary, Spain. Aquatic Botany 46:49–66.

Pulich, W. M., Jr. 1980. Heavy metal accumulation by selected *Halodule wrightii* Asch. populations in the Corpus Christi Bay area. Contributions in Marine Science 23:89–100.

Pulich, W. M., Jr. 1982. Edaphic factors related to shoalgrass (*Halodule wrightii* Aschers.) production. Botanica Marina 25:467–475.

Pulich, W. M., Jr. 1985. Seasonal growth dynamics of *Ruppia maritima* L. s. l. and *Halodule wrightii* Aschers. in southern Texas and evaluation of sediment fertility status. Aquatic Botany 23:53–66.

Pulich, W. M., Jr. 1989. Effects of rhizosphere macronutrients and sulfide levels on the growth physiology of *Halodule wrightii* Aschers. and *Ruppia maritima* L. s. l. Journal of Experimental Marine Biology and Ecology 127:69–80.

Pulich, W. M., and W. A. White. 1991. Decline of submerged vegetation in the Galveston Bay system: chronology and relationships to physical processes. Journal of Coastal Research 7:1125–1138.

SAS Institute, Inc. 1985. SAS/STAT guide for personal computers. Version 6. Cary, North Carolina.

Short, F. T. 1987. Effects of sediment nutrients on seagrasses: literature review and mesocosm experiment. Aquatic Botany 27:41–57.

Simmons, E. G. 1957. An ecological survey of the upper Laguna Madre of Texas. Publication of the Institute of Marine Sciences University of Texas 4(2):156–200.

Sokal, R. R., and F. J. Rohlf. 1981. Biometry. 2nd edition. W. H. Freeman and Co., San Francisco, California.

Trefrey, J. H., and B. J. Presley. 1976. Heavy metals in sediments from San Antonio Bay and the northwest Gulf of Mexico. Environmental Geology 1:283–294.

Van Tussenbroek, B. I. 1995. *Thalassia testudinum* leaf dynamics in a Mexican Caribbean coral reef lagoon. Marine Biology 122:33–40.

Ward, G. H., and N. E. Armstrong. 1992. Ambient water and sediment quality of Galveston Bay: present status and historical trends. GBNEP-22. Galveston Bay National Estuary Program, Houston, Texas.